Chernobyl Radioactivity Impacts and Remediation of Forest Ecosystems

by

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ABSTRACT

The Chernobyl accident in 1986 spewed large quantities of radioactivity into the troposphere. This radioactivity was dispersed worldwide, with areas near Chernobyl being affected the most.

Trees and other biota can absorb/uptake radionuclides when present in their growing media. Hazardous impacts of ionizing radiation usually occur when radionuclide concentrations exceed threshold levels. Based on a limited review, the levels of radioactivity present in forest ecosystems closer to Chernobyl are cited, hazardous radionuclides are identified, their mechanisms of uptake and effects on plants are discussed, and remediation measures are suggested.

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INTRODUCTION

The Chernobyl accident occurred on April 26, 1986, and the emissions spanned over the next ten days in two distinct modes (Medvedev, 1990). This nuclear incident was of staggering proportions, releasing an estimated 50 MCi of radioactivity into the lower atmosphere (troposphere). The core inventories and total releases, taken from the Medvedev book - The Legacy of Chernobyl - are shown in Table 1. Also shown is the distribution of radionuclides onto neighbouring areas in Figures 1 and 2.

The radionuclide releases were in gaseous phase as well as in particulates of differing sizes. Based on half-lives and nuclear core proportions, the radionuclides of primary concern are plutonium radioisotopes ($^{238, 239, 240, \text{ and } 241}$Pu), strontium-90 (^{90}Sr), and cesium-137 (^{137}Cs). Plutonium is essentially an alpha (helium nuclei) emitter, and metabolically is extremely toxic when ingested. ^{90}Sr is a beta emitter (electrons), while ^{137}Cs principally emits gamma rays (electromagnetic radiation, wavelength less than 10^{-3} nm).

The initial explosion of the reactor core released radionuclides in particulate and gaseous phases. The particulates consisted of most radionuclides present in the reactor core, including the transuranic elements such as plutonium radioisotopes. The gaseous phase consisted of radioactive gases, such as krypton-85, xenon-133, and iodine-131, as well as volatile radionuclides (Table 1).

The spread and distribution of radionuclides depended upon their particle size, and the wind direction and velocity prevailing at the time of release (Fig. 1 and 2). An initial plume, which included the gaseous and particulate materials, spread northwest into Byelorussia. A second plume, exacerbated by a core meltdown, spread southwards into the Ukraine, and then east into Russia (RSFSR). The second plume consisted mainly of volatile and semi-volatile radionuclides, such as ^{90}Sr and ^{137}Cs.

Table 1. Core inventories and total releases from the Chernobyl accident.

Element	Half-life (d)	Inventory[a] (Bq)	Percentage released
Krypton-85	3,930	3.3×10^{16}	~ 100
Xenon-133	5.27	1.7×10^{18}	~ 100
Iodine-131	8.05	1.3×10^{18}	20
Tellurium-132	3.25	3.2×10^{17}	15
Caesium-134	750	1.9×10^{17}	10
Caesium-137	1.1×10^{4}	2.9×10^{17}	13
Molybdenum-99	2.8	4.8×10^{18}	2.3
Zirconium-95	65.5	4.4×10^{18}	3.2
Ruthenium-103	39.5	4.1×10^{18}	2.9
Ruthenium-106	368	2.0×10^{18}	2.9
Barium-140	12.8	2.9×10^{18}	5.6
Cerium-141	32.5	4.4×10^{18}	2.3
Cerium-144	284	3.2×10^{18}	2.8
Strontium-89	53	2.0×10^{18}	4.0
Strontium-90	1.02×10^{4}	2.0×10^{17}	4.0
Neptunium-239	2.35	1.4×10^{17}	3
Plutonium-238	3.15×10^{4}	1.0×10^{15}	3
Plutonium-239	8.9×10^{6}	8.5×10^{14}	3
Plutonium-240	2.4×10^{6}	1.2×10^{15}	3
Plutonium-241	4,800	1.7×10^{17}	3
Curium-242	164	2.6×10^{16}	3

[a] Decay corrected to 6 May 1986 and calculated as prescribed by the Soviet experts.

Source: INSAG-1 Report, p. 34.

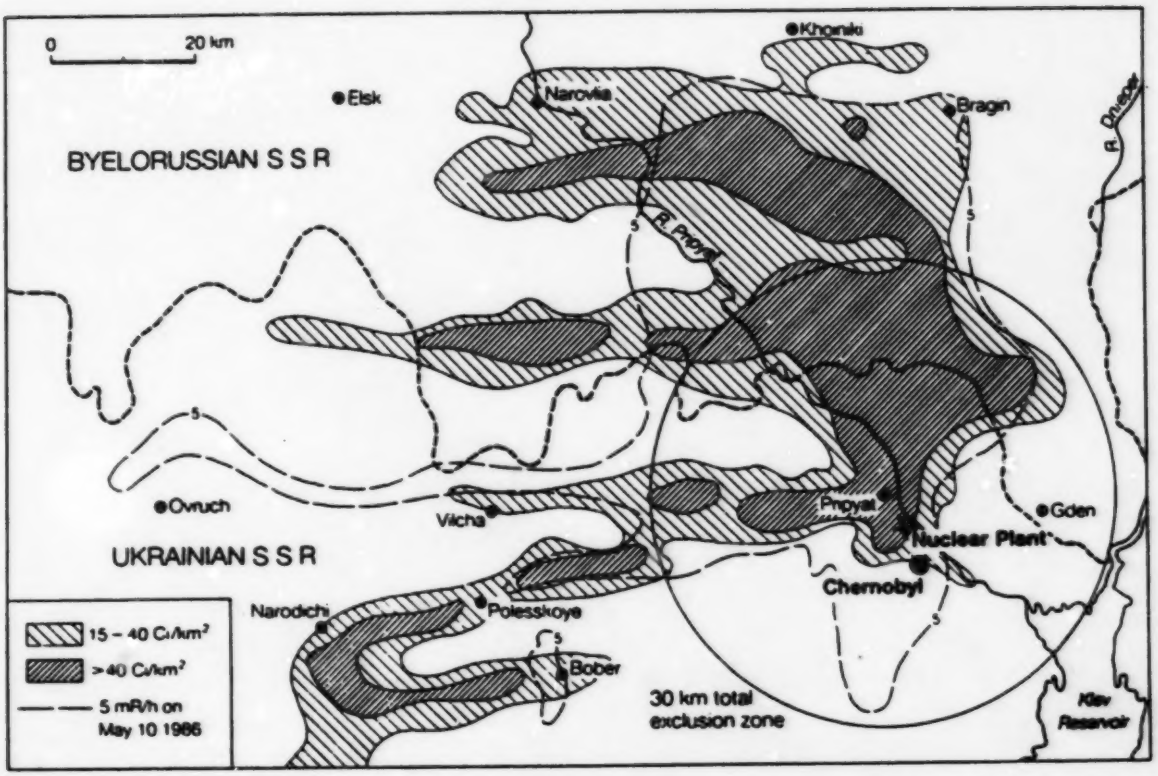

Figure 1. Areas of heavy contamination around the exclusion zone (marked by a 30 km radius circle) with ^{137}Cs as measured during 1988. Only two levels are indicated. The contour marked by isolines indicates the territory which was contaminated above 5 mR/h of gamma radiation on 10 May, 1986. (Source: Medvedev, 1990)

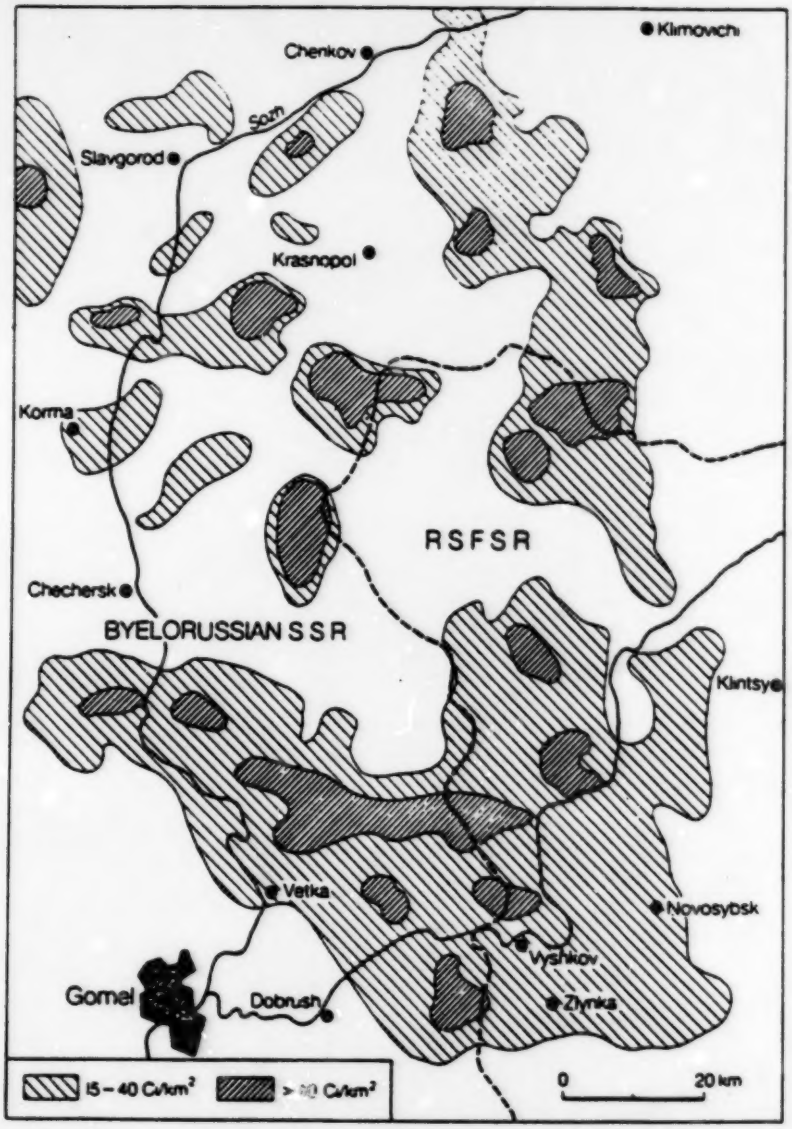

Figure 2. Map of the heavily contaminated districts of Byelorussia and the RSFSR (Gomel, Mogilev, and Bryansk regions). The contamination was by ^{137}Cs and was probably caused by local rain at the end of April 1986. The total contaminated territory (levels above 15 Ci/km^2) is larger than the area in and around the exclusion zone. About 1500 km^2 contained more than 40 Ci/km^2.
(Source: Medvedev, 1990)

RADIONUCLIDES IN FORESTRY ECOSYSTEMS

Initially, the radionuclides were deposited both on the surfaces of vegetation (direct fall-out) and on soils. However, since the retention time on plant surfaces is relatively short, the contaminants have by now been washed down to the soil. Thus, all radionuclides are now either retained by soils or washed into the water bodies.

The deposition of radionuclides in soils is of primary concern to forestry ecosystems. The soil composition is key to the retention/release of radionuclides. The fine particulates (clays) and organic colloids have excess negative charge on their surfaces whereby they can adsorb cations. Also, fine particulates offer a great deal of specific surface area, and both ionic and non-ionic soluble substances can adsorb via physical adsorption mechanisms. There are many other factors which may affect radionuclide retention in a soil. These factors include nuclide concentration and chemical characteristics, concentration of competing ions, soil pH, and concentration of chelating agents (Bocock, 1981).

Plants actively and passively absorb radionuclides via their root systems. However, root uptake is affected by factors such as radionuclide concentration relative to other substances in soil, radionuclide dissolution rates, soil temperature, and management practices, e.g. soil fertilization, ploughing, etc. (Bocock, 1981)

Soils in the Chernobyl area are predominantly clay-based in a temperate region, with a sufficient amount of organic matter (Encyclopedia Britannica). Therefore, these soils are expected to have high cation exchange capacity which will retain cationic radionuclides on their surfaces. Generally, polyvalent cations are more strongly adsorbed by soils than monovalent cations. Thus, it is likely that the deposited hazardous radionuclides, viz. ^{137}Cs, ^{90}Sr, and Pu, have been trapped and retained in the top few centimetres of these soils. Hence, it would seem plausible that more radionuclides are available to forest flora, especially to plants with shallow root systems, as opposed to mature trees with deeper root systems. Nonetheless, organic acids are generated in soils containing organic matter, such that the radionuclides may be dissolved and leached into deeper soil layers for tree root uptake.

All radionuclides have unique physical and chemical characteristics; therefore, their uptake by the plant root systems will be radionuclide-specific (Kathren, 1984) as briefly discussed below.

Strontium

Strontium compounds are quite soluble in water. Therefore, most of the strontium in soils occurs either in solution phase, or on the soil cation exchange complex. Hence, the hazardous radioisotope ^{90}Sr is readily available for plant uptake. The requirement of most plants for calcium is quite high. Also, strontium is very similar to calcium chemically; therefore, the plants can easily mistake strontium for calcium, and absorb it. Strontium is generally more available at lower pH values than at higher. It should be noted that strontium only moves upwards in plants, and therefore collects in shoots and leaves (Bocock, 1981).

Cesium

Chemically, cesium is very similar to potassium. In soils, therefore, cesium will exist in soil solution as well as on the soil exchange complex. Like potassium, cesium can also be fixed irreversibly in clay lattices of minerals like illite.

Potassium is usually quite abundant in soil solution. Therefore, for uptake by plant roots, ^{137}Cs will have to compete with potassium. In certain circumstances where the soil solution composition of potassium is high, the uptake of ^{137}Cs by plant root systems will be retarded due to a common-ion-effect phenomenon. Like potassium, cesium isotopes will be equally distributed in all plant parts.

Plutonium

There is extremely little plant uptake of transuranics, such as plutonium, largely due to insolubility of these elements in soil solution. There is, however, some concern that certain bacteria may solubilize plutonium compounds, rendering them available to plants. In acidic soils, plutonium will be slightly more available than in basic soils because plutonium precipitation may occur at higher pH values (Bocock, 1981).

There is some possibility that rain and wind will splash contaminants onto the base of plants (Pinder and McLeod, 1988). Therefore, during logging and processing operations, this may be a cause for concern.

Despite the low plant uptake of most radionuclides, nonetheless, serious concerns remain due to their extreme toxicities.

TOXICITY OF RADIONUCLIDES

The toxic effects of radioactive contaminants stem from their ionization capabilities. In plants, ionization of atoms in important subcellular structures, such as nucleic acids, can cause cell disfunction/mutations. The transformation of DNA is especially hazardous because of its replication in reproduction. In humans and wildlife, cancers and birth defects are caused by such genetic transformations.

The effects of contamination on a forest ecosystem may not be as visibly pronounced. Plant cells are generally less susceptible to radiation death than animal cells (McKee et al., 1988). Also, some populations have the ability to adapt to chronic contamination such that the population survives (McKee et al., 1988). In such situations, however, individual organisms may not be as healthy or robust. As well, large plants such as trees are more sensitive to initial radionuclide fall-out than

small plants. As well, coniferous are more sensitive than deciduous. About 400 hectares of pine forest in the path of the first plume died immediately as a result of direct fall-out, while hardwoods survived (Medvedev, 1990).

The minimum total radiation exposure required to cause changes in forest community species composition ranged from 3 to 270 Gray (Gy) (International Advisory Committee, 1991). This wide range of values is attributable to different sensitivities of the plant species affected and the variable rates of exposure. Long-term effects of radionuclide contamination on plants include stunting of growth, a general linear decline in productivity, and loss of reproductive capacity (McKee et al., 1988; International Advisory Committee, 1991).

The radiation hazard from radionuclides having short half-lives has decreased considerably since the accident. Therefore, the number of spontaneous mutations in trees has been declining since 1987. Also, there have been no apparent deleterious impacts on trees resulting from contaminant uptake through plant roots. (International Advisory Committee, 1991). The levels of contamination simply may not be high enough to cause widespread

disruptions to forest populations beyond those caused by initial fall-out. Variability of species, soil types, and radionuclide levels make general characterization of hazardous impacts difficult. However, it would appear unlikely that any long-term detrimental effects would develop in plant communities when the maximum absorbed dose rate is 10 mGy/day or less (International Advisory Committee, 1991). Again, the probable lack of stress on forest populations does not preclude genetic or somatic difficulties for individual organisms.

REMEDIATION TECHNOLOGIES

There are several well established techniques of remediation in response to radionuclide fall-out. These technologies are based on two general strategies; outright removal of contaminated materials, and suppression of contaminant ecological impacts.

Except for a few specific strategies, such as clearing of young underbrush, fallen leaves, and fungi with high radionuclide content, removal would likely be ineffective and expensive in a forestry ecosystem. Therefore, concerted effort should be placed on

suppression strategies. These strategies are based on upgrading of soils with the addition of competitive elements. This will reduce the amount of radionuclides taken up by plants.

The Pacific islands of Bikini Atoll was contaminated with radioactivity from atmospheric testing of atomic bombs during the late 1940s and early 1950s by the US military. The local population had to be removed from the islands due to radiation exposure. Since then the remediation techniques of radionuclide suppression have been developed for the reduction of contamination to safe levels at the island (Anderson, 1988). Generally, the key hazardous radionuclides originating from atomic bomb testing and nuclear power plant accidents are very similar. Therefore, the approaches taken at this Pacific Island are deemed relevant and useful for assessing/designing remediation technologies for a forestry ecosystem.

Special competitive elements may be added to soils to suppress radionuclide uptake by plants; some specific effects of the two key such elements (agricultural amendments) are described below:

Potassium

Perhaps the most promising strategy is the addition of potassium (K) to soils. It has been shown that soils with high exchangeable potassium concentrations restrict plant uptake of ^{137}Cs by a factor of upto 3.5. (Anderson, 1988; Bocock, 1981; International Advisory Committee, 1991). However, the addition of excess potassium to soils already high in K content has little additional effect on cesium uptake retardation. Also, the blanket addition of mixed fertilizers is not as effective since nitrogen is known to facilitate plant uptake of ^{137}Cs. Of particular significance for this strategy is the area's abundance of potassium deposits in the form of potash (Derry, 1990).

Calcium

A similar possibility is the addition of calcium (Ca) to contaminated soils. Strontium is virtually indistinguishable from calcium biologically. The Sr/Ca ratio in a plant is reflective of the ratio prevailing in the surrounding soil; generally the two are equal (Bocock, 1981; Kathren, 1984). Thus, increasing the amount of calcium in a soil will reduce the uptake of ^{90}Sr by plants. In fact, extra calcium also restricts cesium uptake, although not as strongly as potassium (Bocock,

1981). Similar to potassium, introducing excess Ca to calcium-rich soils has little additional effect. A possible difficulty with this procedure is the alkaline nature of calcium compounds. However, the uptake of Pu is inhibited by high pH; so this new situation will be beneficial for soils in general.

CONCLUDING REMARKS

Based on limited review, total release of radioactivity and its distribution patterns are described. Also discussed are the nature and mode of action of hazardous radionuclides on forestry ecosystems. This report concludes with possible measures for remediation, based primarily on the Bikini Atoll model.

Presently, there are many studies underway to assess the long-term impacts of radioactivity on forestry ecosystems and the environment at large. However, as of now we have no access to information on the actual nature of soils, the levels and extent of distribution of radioactivity in the ecosystems of interest, and the social and economic values attached to these forestry ecosystems.

RECOMMENDATIONS

If suggestions for in-depth and detailed remediation measures are required, then access and review of this recent information is mandatory. Perhaps, a site visit and discussions with the researchers involved will be of additional help in formulating detailed and realistic remediation measures.

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